Using a data synthesis approach to generate novel results for ecosystem recovery in deer-affected forests

Trillium grandiflorum

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Abstract
Overgrazing by white-tailed deer (*O. virginianus*) has been an ongoing problem in Rondeau Provincial Park since the turn of the century. Since the recent reintroduction of annual culls by park officials, deer densities have declined but the extent to which this control has been effective in restoring plant communities remains to be seen. Such research may only be conducted by synthesizing long-term data sets, dating back to the establishment of artificial exclosures in the park and commencement of culls, and producing novel information. It was determined that *T. grandiflorum* populations within the park are still in the process of recovery, even following eighteen years of experimental exclosures. Thus, recovery from recently reduced deer populations is still underway and occurring at a slow rate. It is also believed that high deer herbivory aids in the spread of invasive species, however this was not found to be the case in Rondeau. Spread of invasive *A. petiolata* is diminishing in intensity and was found to have little effect on species richness in affected areas. These results illustrate the utility of data synthesis principles and practices in long-term ecological studies and stress the importance of process metadata and digital data repositories in furthering such a field.

Key words: *Trillium grandiflorum, Odocoileus virginianus*, overgrazing, herbivory, *Alliaria petiolata*, species richness, exclusion, recovery, data synthesis

Table of Contents

List of Figures

List of Tables

Chapter 1: Introduction and Literature Review

1.1 Measuring Ecosystem Recovery: the Relevance of Long-term Datasets

1.1.1. Use of Large-Flowered Trillium *(Trillium grandiflorum)* in Restoration Ecology: its usefulness as an indicator species for assessing white-tailed deer herbivory

One of the main goals of the field of Restoration Ecology is to measure the success of ecosystem management and to track the recovery of habitats and species. However, this field has seen much debate over how to define the recovery or repair of an existing, damaged habitat. Meffe et al. (1997) describe ecosystem recovery as the aspect of ecological restoration which aims to restore the system to its previous state such that its most desirable attributes are returned (Tanentzap et al. 2009). The use of Indicator Species as evidence of habitat recovery is widespread, and at Rondeau Provincial Park, Ontario, one of the desirable attributes is considered to be the presence of *Trillium grandiflorum*, the province's official plant. In the early 1990s, high densities of white-tailed deer (*Odocoileus virginianus*) had reduced the presence of *T. grandiflorum* to very low densities (Koh 1995). Both at Rondeau, and elsewhere, *T.grandiflorum*, has been used as an indicator of the impact of white-tailed deer (Anderson 1994; Kirschbaum and Anacker 2005; Koh et al. 2010).

T. grandiflorum is a perennial, understorey plant, common to the deciduous forests of the Great Lakes region (Hanzawa and Kalisz 1993). It emerges early in spring, flowers in late April to early May, and has senesced by late July to August. Since it is one of the first woodland plant species to appear, it is a prime green target for white-tailed deer, which browse on twigs in the winter. This early emergence, however, is balanced by the extremely slow growth rate at which leaves and flowers are produced, making the species extremely long-lived - plants may live for over 30 years (Hanzawa and Kalisz 1993). Due in part to its valued anthropogenic role, the presence of *T. grandiflorum* is an ecosystem trait highly valued by humans. However, it is not just passers-by remarking at the beauty of this flower who are interested in its continued presence; ecologists have been looking to the species for decades in tracking recovery of disturbed forest systems (e.g. Koh 1995, Koh et al. 2010).

Species that are most suited for serving as ecosystem indicators, in general, must possess traits that can be easily measured and tracked, be relatively common within an area, and in the specific case of indicators of deer densities, must also exhibit a correlation with trends in deer impact (Kirschbaum and Anacker 2005). *Trillium* may be used as an indicator of understory health and intensity of deer browsing within a given ecosystem (Anderson 1994; Jules 1998; Kirschbaum and Anacker 2005; Koh et al. 2010). As such, the most common *Trillium* species in much of the northwestern USA, southeastern Canada, Ontario and Rondeau Provincial Park is *T. grandiflorum* (Britton and Brown 1970). Its seeds are ant dispersed, which is a reliable, yet inefficient method, as seeds are unable to travel far from their point of origin, effectively limiting the range

of expansion and allowing for continual monitoring of the same *T. grandiflorum* individuals over an extended period of time (Gates 1940, Webster et al. 2005).

Total biomass, leaf area, and leaf length all serve as a function of an individual plant's mean stem height, allowing for the further use of these variables in estimating deer grazing pressure (Koh et al. 2010). White-tailed deer (*Odocoileus virginianus*) in Rondeau Provincial Park, however, express spatial variance, which limits the applicability of these descriptive traits to site-level indicators (Koh et al. 2010). In areas affected by high deer densities and consequent overgrazing, the mean height and leaf area, population density, and individual growth rates of *Trillium grandiflorum* decrease (Anderson 1994, Augustine and Frelich 1998, Rooney and Waller 2001, Kirschbaum and Anacker 2005). While population decline may occur in ecosystems that are not so deer-disturbed, the rate and intensity are not as severe as in deer-overgrazed regions (Rooney 2003).

Kirschbaum and Anacker (2005) claimed that a significant correlation between deer browsing levels and densities, and the indicator variables or metrics measured in *Trillium spp.,* had not yet been established. A statistically significant relationship between deer density and *Trillium* height was presented in Koh et al. (2010). Deer preferentially feed on taller *Trillium* plants, which are unable to grow back to their initial height in the year following grazing, due to their within-season determinate growth (Knight 2003; Koh et al. 2010). An intrinsic feature of *Trillium spp*., is that flowering individuals tend to be greater in height than their non-flowering counterparts, resulting in grazed *Trillium* populations to consist primarily of single-leaved stems and juvenile life stages (Anderson 1994, Kirschbaum and Anacker 2005, Koh et al. 2010). This results in decreased reproductive output of the local *Trillium* population and a shift in its age composition. This relationship was previously demonstrated by determining that changes in the stem height of *Trillium grandiflorum* could be used to extrapolate the health of neighbouring, herbaceous flora (Anderson 1994). Using data spanning a fifteen-year period, Koh et al. (2010) found that following deer exclusion, *Trillium spp.* heights increased in an area, and that this was a recovery process requiring a minimum of four years following the cessation of intensive grazing. However, associated shifts in the composition of local plant communities, in which many native species are reduced by deer herbivory, has led to the concern that of the gaps in forest understorey communities may be filled by non-indigenous, invasive plant species, especially garlic mustard, *Alliaria petiolata* in the study area of Rondeau Provincial Park (Pearl et al. 1995).

Intensive deer grazing influences species richness and life history traits contributing to the reproduction and spread of herbaceous plants (Kirschbam and Anacker 2005). By reducing cover in these areas, it is also likely, therefore, that deer grazing is effectively decreasing the level of vegetation biomass in high deer density areas, which consequently decreases species richness of an area, in both flora and fauna (Kirschbaum and Anacker 2005, Tanentzap et al. 2010).

It is well-established that grazing by high white-tailed deer densities lowers the effectiveness of reproduction in *Trillium spp.* and also limits its rate and degree of recovery (Augustine and Frelich 1998; Webster et al. 2005; Knight 2003). While deer

grazing on *Trillium* plants removes all foliage, flowers, and fruits, effectively eliminating any chance of regrowth within the same growing season (Augustine and Frelich 1998), *Trillium* are resilient to grazing over an initial period of 1-2 years, allowing for rapid recovery if deer densities are reduced within this period (Augustine and Frelich 1998). However, when high deer numbers persist for at least 5 years, the negative effects of intense herbivory become increasingly evident and more difficult to reverse. At this point, the effects exerted are not only cumulative in a linear sense, but also accelerate because they affect future population levels, by their impact on reproduction, namely lowered seed set through a severe reduction in floral density (Augustine and Frelich 1998; Knight 2003; Webster et al. 2005). In particular, the long generation time of *Trillium spp*. restricts the speed at which populations of the species can recover to their former density, following the reduction of grazing pressure (Augustine and Frelich 1998).

High deer densities have been present in Rondeau Provincial Park since the turn of the century (Killan 1993), increasing the difficulty of managing for *Trillium* populations to bounce back. This problem is compounded by the fact that, even in light of lowered *Trillium* densities, these species continue to be grazed upon with high intensity, while simultaneously competing for resources with better adapted, graze-tolerant species (Augustine and Frelich 1998; McLachlan and Bazely 2001; Tanentzap et al. 2010). As a result, shifts in plant community composition have occurred, whereby species more resilient to browsing have increased in abundance following periods of extended overgrazing (Koh 2002). It has been proposed that if vulnerable plants were to be transplanted to an area during a period of lowered deer density, the effect of deer on their survival would be minimal (Augustine and Frelich 1998). This, however, would require active and intensive management on the behalf of park officials and botanists.

1.1.2 Tracking overgrazing by white-tailed deer

High white-tailed deer, *O. virginanus,* densities occur in numerous sites across Canada and the eastern United States, and other deer species have also reached high densities in other parts of the globe, including New Zealand (*Cervus elaphus*) and Japan (Coomes et al. 2003; Wilson et al. 2006; Tanentzap et al. 2009; Takatsuki 2009). At the Smithsonian's Conservation and Research Center in Washington D.C., regulations barring deer hunting on centre grounds were abolished, due to a rapid increase in *O. virginianus* population size. This increase resulted in a greater incidence of disease and a higher extinction risk for highly sensitive, native deer species (McShea and Rappole 1997). Following the reintroduction of deer hunts, animal rights groups organized opposition campaigns that effectively put an end to all organized future cullings of *O. virginianus* on centre grounds (McShea and Rappole 1997). Similar issues have arisen in southwestern Ontario, where activists are currently at odds with indigenous groups and park management officials over hunting practices in the area, and the deer culls that currently occur annually.

In the case of the Smithsonian's ongoing battle with animal-rights activists, the centre looked to alternative methods of deer control, including deer drives, relocation, and most notably, immunocontraception, all of which have proven quite ineffective (McShea and Rappole 1997). The results of this study, then, will also have social

implications, as animal rights groups continue to fight for an end to the annual deer culls in southwestern Ontario parks. By determining how deer herd reductions are affecting native *Trillium spp*. and overall species richness and community composition in previously overgrazed habitats, scientific and quantitative contributions may be integrated into future park management policy for the region.

Several studies have attempted to pinpoint an ideal density for *O.virginianus* populations in deciduous forests at which forest regeneration will occur (Tilghman 1989; Anderson 1994; Pearl et al.1995; Ontario Parks 2001). Systems in northeastern Illinois are believed to regenerate at densities of 4-6 deer per square kilometre (Anderson 1994), while in northwestern Pennsylvania, a deer density of approximately ≤6.95 is considered to allow forest regeneration (Tilghman 1989). Since deer are both grazers (eating leaf tissue) and browsers (eating woody twigs), not only do they remove green biomass but they also cause sapling and shrub death by clipping of the twigs that will bear the following year's growth (Pearl et al. 1995). Before regular culls were initiated, populations in Rondeau were nearing 30-45 individuals per square kilometre, while the park, in its entirety of 32.54 km^2, is only able to hold a population of no greater than 65 to 75 deer, or 6-7 individuals per square kilometre (Pearl et al. 1995; Ontario Parks 2001). During periods of early recovery, however, management officials have suggested reducing deer densities to below this threshold to allow vegetation recovery to occur; following an adequate level of restoration, the number of deer culled may be reduced to meet these guidelines (Ontario Parks 2001). In similarly affected regions of the northwestern United States, ideal densities were estimated at 4-6 deer per square kilometre, a value which, with proper monitoring and control methods, was feasible within the park of study (Anderson 1994).

1.1.3 Interactions between deer herbivory and the spread of invasive species

By decreasing the level of vegetation biomass and lowering biodiversity, high herbivory by white-tailed deer may also increase the likelihoods of invasions by other plant species and provide an opportunity for them to establish and dominate in a highly disturbed system (Webster et al. 2005; Knight et al. 2009; Tanentzap et al. 2010). Newly introduced species are not only highly tolerant of disturbance, but also hardier and more difficult to eradicate, even following lowered deer densities (Tanentzap et al. 2010).

Although deer herbivory may play a role in facilitating the establishment of non-indigenous plant species, these invaders may not necessarily act to inhibit growth of native plants. In a study examining the effects of invasive honeysuckle on *Trillium*, it was determined that, although deer herbivory was associated with the spread of this species, it exerted no negative effects on *Trillium* growth (Leege et al. 2010). A similar research question has been posed for Rondeau Provincial Park, with regards to invasive *A. petiolata*. Although garlic mustard is often considered as highly invasive due to its supposed allelopathic effects on neighbouring native species, in which ectomycorrhizal relationships are disturbed, field studies have failed to support these greenhouse-based observations, over an extended period (Stinson et al. 2006, Vaughn and Berhow 1999; Roberts and Anderson 2001). Studies relying primarily on greenhouse experiments (Stinson et al. 2006), are insufficient in demonstrating naturally occurring phenomena,

while those testing garlic mustard extracts (Vaughn and Berhow 1999; Roberts and Anderson 2001), are similarly inadequate when assessing real-world response. One such field study, in New England, found that the inhibitory effects of garlic mustard on ectomycorrhizal fungi occurred no more than 10cm away from the edge of garlic mustard patches, and also failed to affect total root biomass (Wolfe et al. 2008). Furthermore, Burke (2008) discovered that garlic mustard does not significantly affect the mycorrhizal fungi of *T. grandiflorum*, suggesting that suppression of such fungi by *A. petiolata* may actually be species-specific. Such results demonstrate the challenge of extrapolating laboratory manipulations and experimentation to the field, and may lead to premature conclusions that are inapplicable in real systems.

1.1.4 The need for long-term studies in studies of ecosystem recovery

One approach to monitoring recovery in an area with historically high deer impacts is to lower deer numbers and observe the long-term response (Kirschbaum and Anacker 2005). This is most effective in an area in which permanent experimental exclosures have been established, so as to compare herbivore-excluded patches from the surrounding control regions that experience the full range of environmental conditions. This was the approach taken in the present study, with *Trillium* densities monitored over a period of eighteen years.

Valiela et al. (1989) broadly categorized the types of studies in which long-term data are necessary: of these, a study on *Trillium* response (and forest recovery, as a whole) to deer exclusion falls into the second major class, that of monitoring "responses to external change by long-lived components of the system" (Valiela et al. 1989). With respect to a species as long-lived as *T. grandiflorum*, there is a need for long-term studies in order to avoid making erroneous, premature conclusions (Franklin 1989). Interspecific interactions may also require long-term monitoring in order to elucidate the true nature of such relationships. Franklin (1989) suggests that this is necessary to determine whether relationships shift from being of one variety (e.g. competitive) at the onset of a study, to another (e.g. mutualistic) further down the line, and to examine potential changes in the strength and importance of such associations. This approach is useful at Rondeau, where it has been proposed by park managed that *A. petiolata* may be excluding native species and suppressing their recovery. Long-term research would help to determine whether this is the case, after an extended period, or whether garlic mustard simply develops a more mutualisic, less competitive relationship with its neighbours. Such an interactional shift would result in an initial decrease in species richness, only to have diversity levels stabilize over some time. A key element in designing long-term studies is to keep them simple. Too often, researchers create elaborate experiments that are difficult to maintain over an extended period. By drawing up an experimental design in which the methods and measurements are both clear-cut and easily repeatable, long-term monitoring is not only be feasible, but informative as well (Franklin 1989).

Government-managed parks are considered the most suitable sites for long-term studies, as their protected status and, for the most part, less/un-disturbed ecosystems allow for extended monitoring with little to no interference (Parson 1989). As a result, studies that run in conjunction with government agencies are preferable, as they provide

large areas of land in which researchers are assured that their set-up will remain undisturbed (Franklin 1989). The results of such studies are also of interest to these agencies, in developing and implementing updated management policies. Working with such agencies, however, does not ensure continuous funding for these long-term studies.

Caraco and Lovett (1989) propose several approaches to long-term studies: among them, direct and retrospective methods. The latter relies primarily on old human records, including field notes and old data obtained by prior researchers (Caraco and Lovett 1989).

1.2 Integration of Data Synthesis

Data mining is often cited in the field of computer science, but its applicability also extends into the realm of biology, albeit under the slightly modified umbrella of 'data synthesis.' The key elements of both terms are similar: both attempt to make sense of the data-rich, information-poor world in which we live, via the collection of pre-existing data and the subsequent creation of new databases (Han and Kamber 2001). There is an abundance of data available, but what do researchers actually know about it? What does it all mean? Oftentimes, there are patterns hidden within data, which may only be extracted by compiling large amounts of this unused data by way of an archaeological-like endeavour. In data mining, the unearthing of this information is performed by algorithms, while data synthesis relies more so on active reasoning and decision-making by researchers at each step of the process. That is, data synthesis is the assembly of available data sets into one or more original sets, to which statistical models may be applied in order to test a novel hypothesis (Ellison 2010). Data synthesis is becoming a common trend in ecological papers that seek to answer questions spanning a wide range of space and time (Ellison et al. 2006). Climate change research is an example of such a field in which data synthesis approaches may prove useful, through the integration of such retrospective data and live, streaming data (Caraco and Lovett, 1989; Ellison et al. 2006). Furthermore, Ellison et al. (2006) claim contributions from data synthesis studies currently stand as the most valuable sources of information, in the establishment of policy at all managerial and governmental scales.

Ellison (2010) mentions that one of the primary issues with ecological studies is the lack of reproducibility, as also the case in other scientific fields. The results of field studies are specific to the time and place in which they were carried out; however, by using pre-existing data accompanied by a clear explanation of *how* the novel data set was assembled, manipulations become more repeatable and reproducible (Ellison 2010). Reliable data sets are those that have undergone rigorous processing, including quality assurance (data checking) and quality control (data cleaning) (Ellison et al. 2006). The novel data sets that are produced, however, are only deemed valuable when accompanied by process metadata –a clear idea of how the data were selected and collected, in addition to any judgment calls made by the research in the process of compiling the novel file (Ellison et al. 2006).

The National Centre for Ecological Analysis and Synthesis was established in 1994 at the University of California Santa Barbara, with the goal of using existing data to address major issues in ecology and contribute to sound management policy. With such an institution comes freely accessible long-term data. Since the inception of the centre's public data repository, more than 1800 publications have been produced in collaboration with the NCEAS, including key literature on climate change (NCEAS 2011). This repository allows for the use of long-term data sets by an entirely new generation of researchers wishing to investigate ecological issues which date back to their diaper days. Long-term data sets may span a researcher's entire career, without providing any solid conclusions; by synthesizing data from differing research periods of several investigators, however, it is possible to obtain long-term data whose range pushes human constraints. Such an approach of searching for general patterns in existing data is touted as a particularly invaluable means of predicting ecosystem response to disturbance and in looking at invasive species dynamics (NCEAS 2011).

1.3 Study Objectives

While the value of long-term studies and datasets in ecology and beyond, has long been recognized, the reality of funding for basic research, is that financial support for long-term monitoring has usually been lacking (Likens 1989). Government research councils usually prefer to fund basic research that addresses novel questions and hypotheses, rather than pay for long-term data collection and archiving. When Dawn Bazely and her students, commencing with Saewan Koh, began research at Rondeau Provincial Park in 1991, with his York University B.Sc. Honours Thesis, they had no idea that these field plots and data would still be of interest and value two decades later, to both the scientific community and land managers, alike. The research questions that were posed envisaged timelines of 1-5 years, rather than 20 years. Furthermore, a major criticism from the reviewers, of a 2010 manuscript submitted by Koh et al. was the lack of long-term field data on *Trillium* densities (D. Bazely, personal communication). Of all the studies performed in these plots over the years, none had directly aimed to link changes in *Trillium* density to deer herbivory, because of the long life span of *Trillium*. Additionally, long-term studies of plant community composition responses by different students, spanned 5-6 years, and in the mid-2000s, the park research budgets for supporting the field work declined. By 2009, however, in the midst of reviewing the park's management plan, officials feeling the heat from animal rights activists raised questions with regards to the efficacy of deer culls in forest recovery. In order to address such concerns, long-term studies were necessary, but the proper funding had been lacking in years prior. Motivated by the overhaul of this management plan, officials contributed funds in order for studies to re-commence at these plots, within the summer of 2009.

This study aims to demonstrate and evaluate the utility of data synthesis in an ecological study that looks at the recovery of *Trillium grandiflorum* and species diversity, and the spread and effects of garlic mustard on species richness, following high deer density reduction in Rondeau Provincial Park. The data synthesized come from different studies carried out by different students in the same field plots. The primary objective of the paper is to determine how populations of the indicator species, *T. grandiflorum*, are

responding to the exclusion of white-tailed deer in the park. This was achieved by using past data to generate novel data on *Trillium* plant densities in plots with different grazing histories. At the present time, park species should either be in the early stages of recovery, or headed thereabouts, following the reintroduction of deer culls, after a hiatus from the 1970s, in 1993, and annually, in 1997. *Trillium* population densities and flowering rates, should have were predicted to have increased.

Additionally, there is a widely-held view in the ecological literature that high deer herbivory also facilitates the spread of invasive species, such as *Alliaria petiolata* in the case of Rondeau Provincial Park. By examining garlic mustard densities in areas exposed to historically intensive grazing, but currently in the process of recovering, it is possible to determine whether this introduced species is still spreading, following the management of deer herds. It can also be established whether garlic mustard indeed affects species richness in invaded areas. This involves two main objectives:

1. To determine whether there was an increase in the spread of *A. petiolata* among study sites.from 1994-2009.

2. To determine the nature of the relationship between garlic mustard densities and plant species richness.

Chapter 2: Methods

2.1 *Trillium* Recovery Following Deer Exclusion

Long-term deer exclosures were established in Rondeau Provincial Park in 1978, for the purpose of comparing vegetation growth and recovery in areas of artificial deer exclusion, with the overall forest where grazing was allowed to continue, since previous deer herd controls were stopped during the 1960s (Yaraskavitch 1983). From 1978 and during the 1990s, many other study objectives were fulfilled at these sites; however, at the present time, there is a unique opportunity when the data collected through all of these studies may finally be synthesized to produce a long-term data set that addresses one of the most important questions of habitat management and recovery - namely, whether the reintroduced deer herd management in the 1990s has resulted in increased populations of native plant species.

2.1.1 Study site description: Rondeau Provincial Park, Morpeth, ON.

Rondeau Provincial Park (42°17′N 81°52′W) is a 3,254 hectare sandspit located on the northern shore of Lake Erie (Figure 2.1), within the County of Chatham-Kent (Bartlett 1958; Ontario Parks 2001; Friends of Rondeau 2007). Variations in the peninsula's soil moisture are evident, with the driest areas located on the eastern side of the park (Bartlett 1958). Due to its position in the Carolinian zone, however, the number of days in Rondeau's growing season is estimated to range from 203 to 213 (Bartlett 1958). The park is home to the southern deciduous forest that once spanned the entire region surrounding Lake Erie: its beech and maple populations are mere remnants of the area's ecological past (Clarke 1944, Bartlett 1958). As such, the ecological importance of the park was stressed as early as 1944, while high white-tailed deer populations have been having noticeable impacts on forest structure since the turn of the century (Clarke 1944). White-tailed deer were not initially found in the area, but rather, were introduced by Rondeau's first superintendent, Isaac Gardiner, in 1898, four years after the official establishment of the park (Killan 1993). A handful of deer were brought in from Algonquin Provincial Park and kept in pens, but they quickly escaped and increased to high numbers owing to a lack of natural predators in the area (Bartlett 1958, Killan 1993). In the three-year period spanning 1909 to 1912, the number of deer within Rondeau more than doubled from 150 to 400, resulting in the park's first deer cull the same year (Killan 1993). As of 1944, visible regrowth in understorey communities was attributed to the massive culls of the prior years (Clarke 1944). However, up to this point, the plant communities in areas affected by high deer densities had already shifted to become more representative of grazing tolerant trees and shrubs (Clarke 1944). Nevertheless, deer populations continued to be controlled, on and off, for the next few decades, until 1974, when protests from animal activists caused cessation of the culls (Bartlett 1958; Ontario Parks 2001). Only in the 1990s years has this ban been lifted to allow for control of the extremely high deer populations of Rondeau Provincial Park.

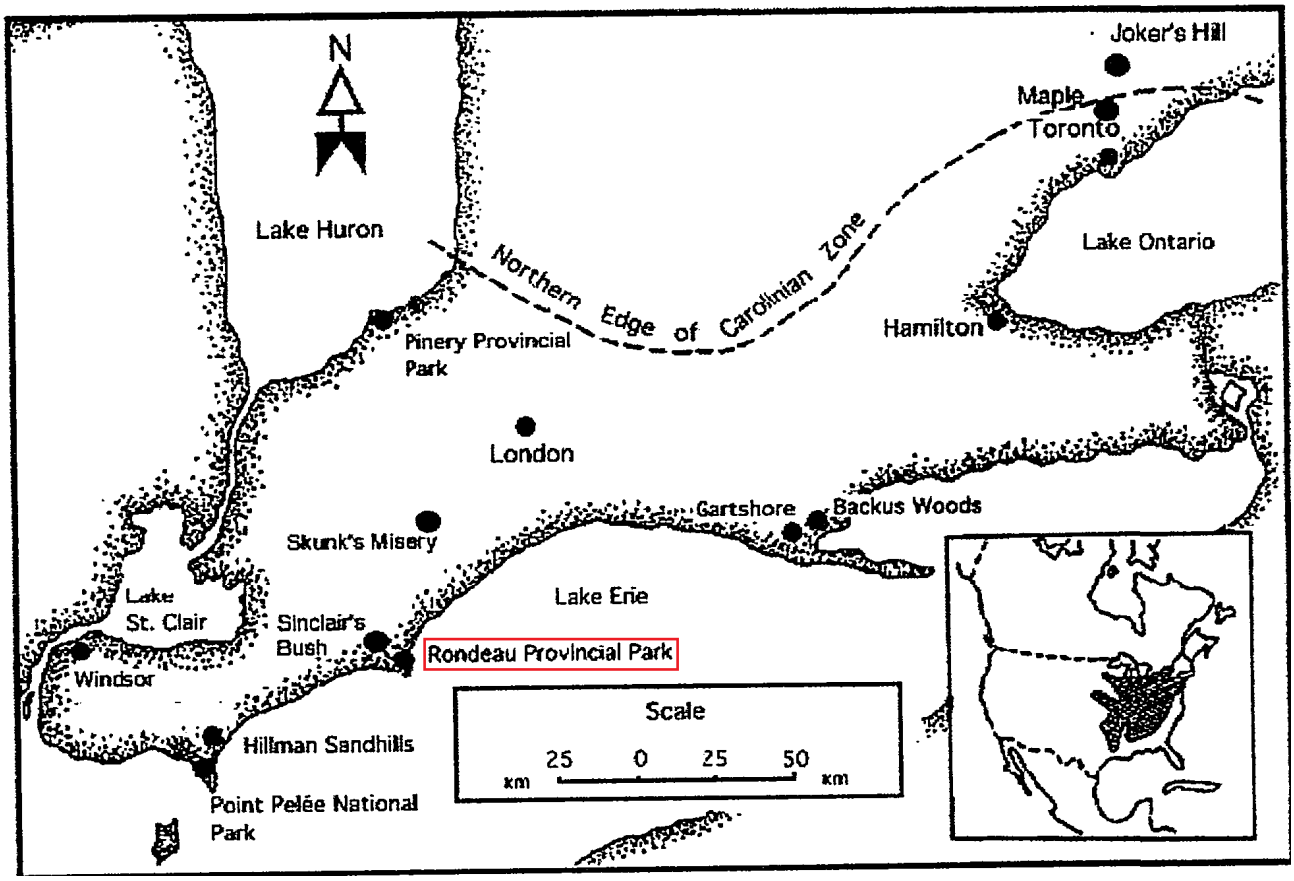

Figure 2.1 – Location of Rondeau Provincial Park in southwestern Ontario, within the Carolinian Zone, also shown, with the inset representing the entire North American range of the Carolinian Zone (Koh 2002; Tagliavia 2002).

2.1.2 Historical Experimental Setup

Long-term exclosures were established in Rondeau Provincial Park in 1978, hereafter referred to as the Bennet and Gardiner exclosures (BEX: 50m x 40m; GEX: 60m x 40m). Ten smaller, permanent exclosures of 1m by 1m were established within each larger exclosure in 1991, with corresponding control sites located just outside of the larger BEX and GEX boundaries. Both Bennet and Gardiner sites each contained 10 exclosures and 10 grazed plots. Also in 1991, a total of 20 stations were established along six 1000m transects representing each of the wet (W1, W2, W3) and mesic (M1, M2, M3) soil zones (Figure 2.2). Four stations were located at M1, and three each at M2 and M3; a similar set-up occurred along the three wet area transects. Each station was set up so as to contain a north and south 2 m x 2m exclosure, as well as a centrally located 1 m x 1 m control (grazed) plot, for each. Twenty exclosures (NEX), $4m^2$ in size, were established in each soil zone, of which exactly ten were at the north-end of the station and the remaining ten, at the station's south. Ten control, grazed plots (RON: $1m^2$) were also established in the same year, in each of the wet and dry soil zones. Within each 2 m x 2 m exclosure, a permanent 1m by 1m plot was designated. The number of exclosures surveyed in any study year, and the variables measured, however, varied depending on the goal of the student's study. Over time, some of the original exclosures were destroyed

by trees falling through them, or became hard to find, as vegetation recovered post-deer herd reductions (Table 2.1). The relative locations of these exclosures within the park are represented by blue points in Figure 2.3, with the Bennet and Gardiner 1978 exclosures circled in red. It should also be noted that another 10 exclosure stations were established along 3 transects in the drier soil areas to the east of the park (Figure 2.2), but due to the high impact of deer browsing on the tree and shrub communities, this area became so open, that it is now managed as an oak savanna habitat and is burned (D. Bazely, personal communication). The understory plant community never contained any *Trillium* plants since the early 1990s.

In 1994, as deer herd reductions were re-introduced for the first time in 20 years, one exclosure at each station was randomly selected and moved 5m south or north from its original location, consequently resulting in the establishment of some additional control plots. A summary of these changes is found in Table 2.2. From this point on, the new control plot was deemed C and the original control plot, formerly referred to as C, was renamed south control (SC) or NC, to represent its nearness to the 1991 un-moved (stationary) southern or northern exclosure (Figure 2.4). Thus, all C values up to, but excluding 1994, were recorded as such, but beginning in 1994, all C values recorded represent the densities of the newly established control plots. Conversely, if the southern exclosure was moved in 1994, a new control was established in some instances, and the original control (C) subsequently referred to as north control (NC). Three new control plots, one NC and two SC, were established for each soil type (Table 2.2).

Trillium grandiflorum leaf and flower heights were measured only in certain years (Table 2.3). Leaf height was considered as the distance from ground level to leaf whorl base (Koh 1995; Firanski 2003), while flower height was similarly measured, to the base of the flower head.

In addition, the frequency of deer culls has been documented in several papers and has been known to occur annually since the winter of 1997/1998, following an initial culling in November of 1993 that resulted in the eradication of 320 individuals (Koh 1995; Koh 2002; Firanski 2003).

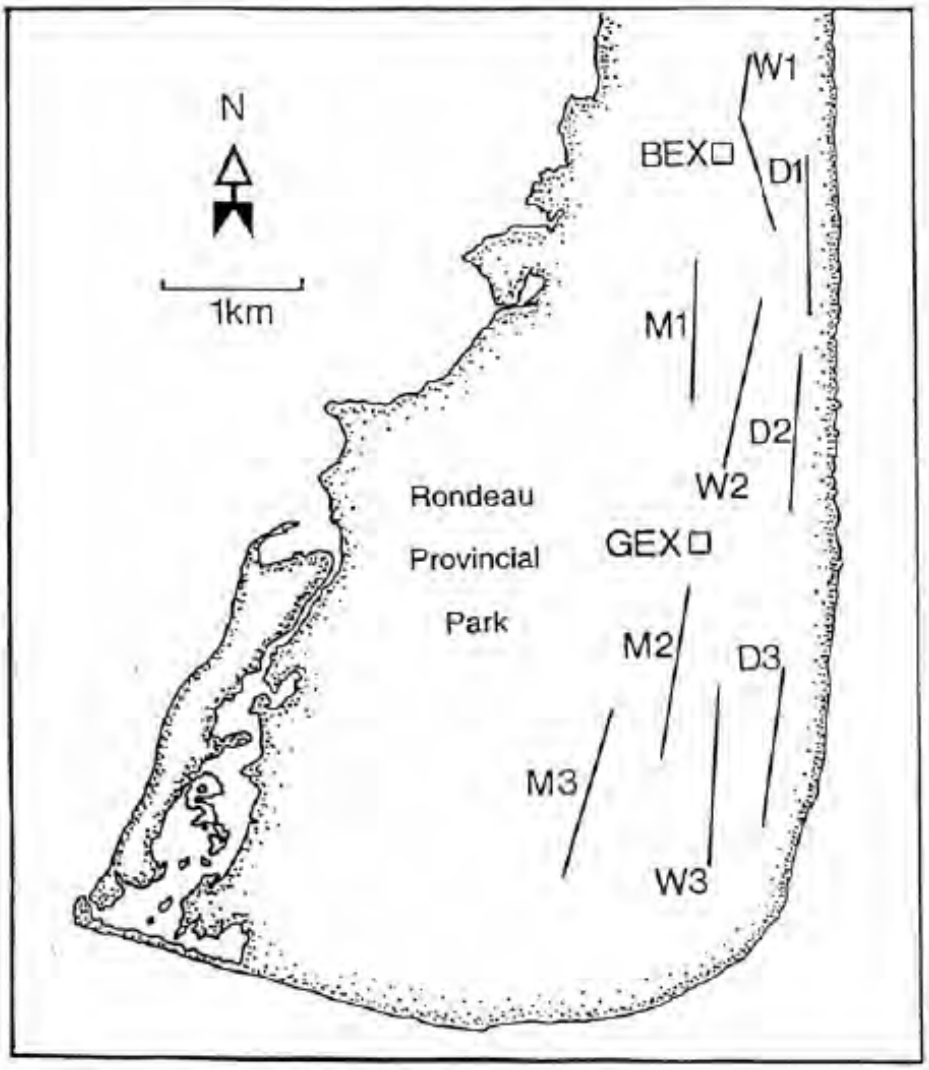

Figure 2.2 - Distribution of the three transects corresponding to each soil type, wet (W1, W2, W3) and mesic (M1, M2, M3), within Rondeau Provincial Park. Also noted are the relative locations of the Bennett (BEX) and Gardiner (GEX) exclosures, established in 1978 (Koh *et al.* 1996).

Table 2.1 - Summary of the respective number of mesic and wet plots surveyed in each year of density counts, and whether the Bennett and Gardiner exclosures observed were representative of the true, 1978 plots, or novel, moved sites.

	# Mesic Plots Sampled	***# Wet Plots Sampled***	***Original Bennett and Gardiner Sampled?***
1991	30	30	✓
1992	30	30	✓
1994	30	30	✗
1995	22	9	✗
1996	32	33	✓
1997	32	33	✓
1998	22	16	✓
1999	30	21	✓
2000	12	6	✗
2001	33	22	✓
2009	32	33	✓

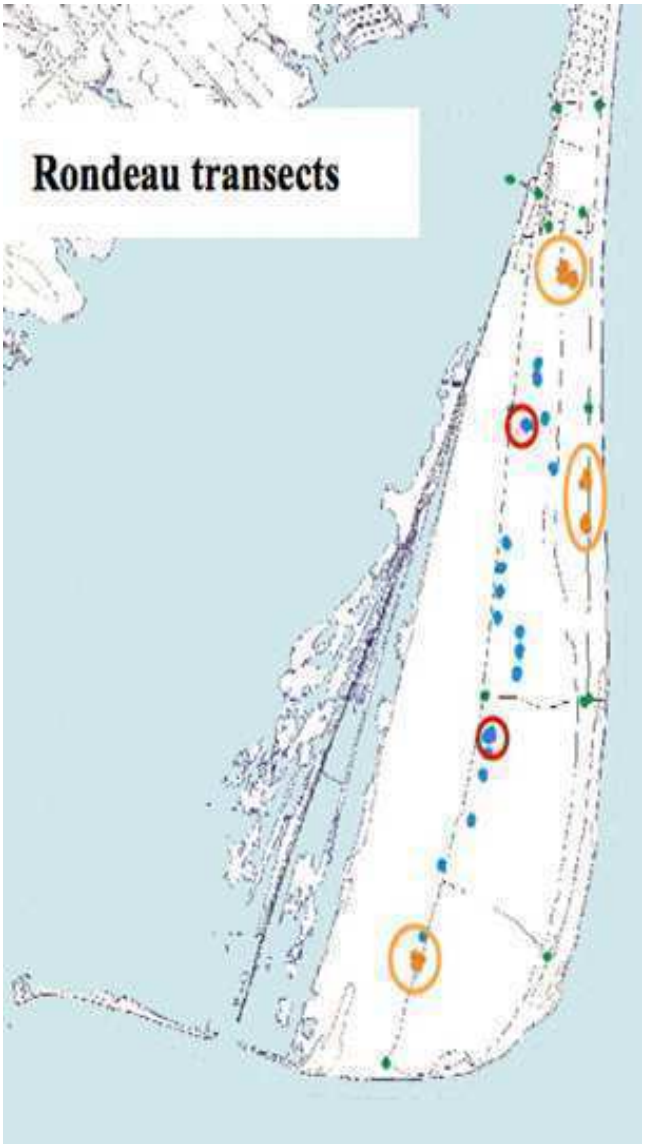

Figure 2.3. - Summary of the locations of deer exclosures, for use in *Trillium* monitoring studies, as represented by blue dots, with the oldest exclosures, Bennett and Gardiner, circled in red. Also visible are the garlic mustard monitoring sites in orange, with the northernmost cluster representing Ponybarn, the southernmost, Southpoint, and the intermediate, Lakeshore (Bazely *et al.* 2009).

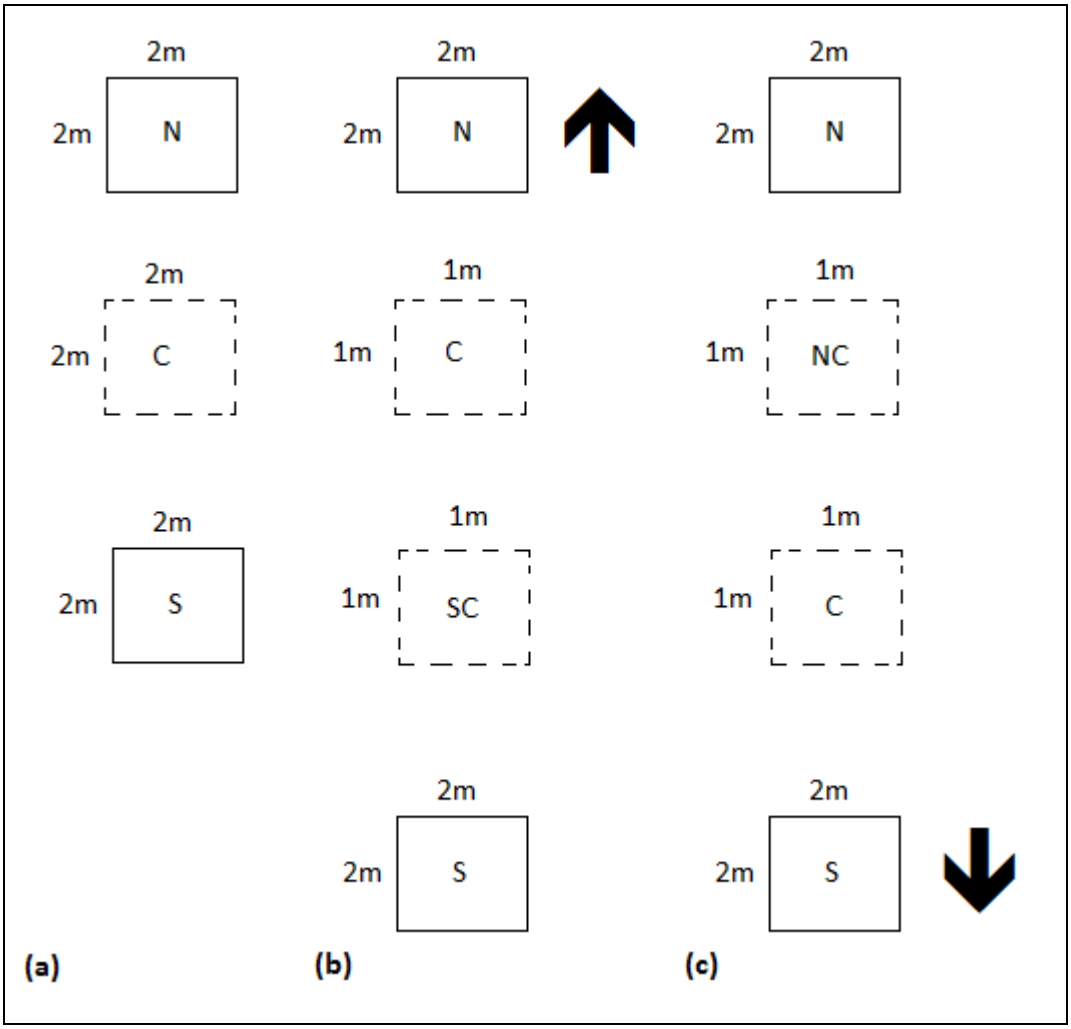

Figure 2.4 – **(a)** Summary of the original, 1991 experimental setup; changes following the movement and reestablishment of **(b)** a northern plot (N), resulting in a newly established south control (SC), and **(c)** a southern plot (S), resulting in a newly established north control (NC).

Table 2.2 – Summary of the north and south plots moved in 1994, and the controls established in the same year.

Station	***Plot***	***Year Moved***	***Year Established***
M1.1	S	1994	
M1.2	N	1994	
M1.3	S	1994	
M1.4	N	1994	
M1.4	SC		1994
M2.0	N	1994	
M2.0	SC		1994
M2.1	S	1994	
M2.3	N	1994	
M3.1	NC		1994
M3.1	S	1994	
M3.4	S	1994	
M3.5	N	1994	
W1.0	N	1994	
W1.1	N	1994	
W1.3	N	1994	
W1.5	N	1994	
W1.5	SC		1994
W2.3	N	1994	
W2.3	SC		1994
W2.4	N	1994	
W2.5	NC		1994
W2.5	S	1994	
W3.1	S	1994	
W3.4	S	1994	
W3.5	N	1994	

Table 2.3 – Summary of all years in which leaf and flower heights were measured, allowing for determination of percentage of flowering *Trillium*.

	1991	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2009**
Leaf Heights	✓	✓	✓	✓	✓	✗	✗	✗	✗	✓	✓	✓
Flower Heights	✓	✓	✓	✓	✓	✗	✗	✗	✗	✓	✓	✓

2.1.3 Application of data synthesis principles and practices

In reconstructing a long-term data set for *Trillium* densities, and percentage of flowering (reproducing plants) at the various Rondeau sites, the approaches used in the fields of data mining and data synthesis were adopted. This consisted of locating as many electronic data sets that were available, as well as locating original field notes of students. In doing so, field data on *Trillium* was found in past field notes that had never been entered and used by students. Some of the electronic data sets were in versions of past statistical programmes that could not be opened on current computer systems, and these files had to accessed by different means. The source of these data sets varied. Some were from experiments that had vastly different study objectives, including the effect of shade/lighting on species growth, *Trillium* height as an indicator of deer browsing, and work on *Arisaema triphyllum*. The collection and entry of field data in each year was performed by a number of different individuals (Table 2.4). However, by combining all of these data, that were in some cases, incidentally collected alongside data that was directly relevant to the studies at the time, a novel set of long-term data for *Trillium* density and percentage of reproductive individuals could be created. It was then possible to apply this data set to an entirely different study question, one that researchers had not explicitly sought to answer in their past field studies.

Once the different datasets were identified and accessed, the second step in this data synthesis process was cleaning the data to be used – that is, ensuring the accuracy of data entry, and investigating further and removing any noise or inconsistencies that may be present (Han and Kamber 2001). These data were then integrated, from all of the different sources.

Trillium densities for 1991 were obtained from A.J. Tanentzap, as used in previous studies, while the percentage of flowering *Trillium* was determined by converting outdated CricketGraph and Statview files to modern Excel .xls format, via an external CD-ROM drive running on Mac OS 7. This new output file also provided the height to leaf and height to flower measures for each plant surveyed, which were then converted into a measure of percentage flowering *Trillium*, with the Bennet and Gardiner exclosures serving as one category, and the wet and mesic exclosures, a second.

Trillium densities for 1992 were translated from field notes to an Excel spreadsheet by Tracy Tanentzap, while the percentage flowering *Trillium* was calculated, as above, using information from a pre-existing spreadsheet, entered by Harpreet Atwal. The same file was used for 1993 heights; however, no measure of true density was available for this year. Densities for 1994 were extrapolated from counting the number of plant heights measured at each station, which also provided the percentage of flowering plants. Note that although exclosures were moved in this year, the measures taken are reflective of the 1991 exclosures and not the new ones, as noted in the field notes. In addition, both the Bennett and Gardiner exclosures measured in this year are not the original plots, but rather were re-established and organized differently (for a summary of all years in which the true Bennett and Gardiner exclosures were surveyed, refer to Figure 2.1). It was still possible to obtain a measure of percentage of flowering *Trillium*, however; as arbitrary plants were measured in each year, and not the same

individual over a number of years, the selection and inclusion of heights should not be specific to a particular plot. Thus, these measures, taken in the general vicinity of the original Bennett and Gardiner exclosures, were deemed to meet the general criteria for inclusion in a *Trillium* density dataset. For 1995 densities, a similar method of extrapolating from the number of height values was used, which then provided the subsequent percentage measures. Once again, a novel set of BEX and GEX sites were surveyed. For the years of 1996 to 1998, inclusive, densities were entered by Tracy Tanentzap, while flowering heights were not measured. Density counts were taken on multiple dates; however, the highest values were included in the refined data set.

In the case of 1999, all that could be recovered was a percentage cover matrix, which included not only *Trillium*, but all species present and accounted for. The original field data sheets in which plant frequency as well as percentage cover of different species, could not be located. In order to relate these covers to density values, it was necessary to establish a relationship between the two parameters. This was achieved by performing a quadratic regression and bivariate correlation analysis in SPSS version 19.0, on the 2001 data, obtained by Carrie Firanski and provided by A.J. Tanentzap. In addition to both percentage cover and density measures being easily obtainable for 2001, it was also a year in which the original Bennett and Gardiner exclosures had been measured. Thus, by regressing cover on density for each station, for which both measures were available, the resulting equation was used to estimate 1999 density values from percentage cover data.

Another unexpected data finding occurred for 2000, in which a maximum of 10 plants were measured in each wet/mesic plot, only for height. Since not all plants in a plot were measured for their height, beyond the first ten plants, these data could not be used for density calculations. Therefore, any plots in which exactly 10 plant heights were given were excluded from the density summary table, and listed as not sampled (NS), while plots which had fewer than 10 heights measured were used to generate density data, because they represent data for all plants in a plot. In calculating the percentage of flowering *Trillium*, however, all values for flowering and non-flowering plant heights could be used. Once again, the BEX and GEX locations used were not reflective of the original 1978 exclosures; however, their data was used in the latter calculation. The most recent set of data was entered by a Research at York student, Pratyusha, who had entered the heights of all surveyed plants, which was used to determine density. In addition, the original BEX and GEX plots were counted.

Values of *Trillium* densities in all of these years were integrated to produce a new, primary data set, from which several subsets could be isolated and analyzed. By reorganizing based on the year of exclosure establishment (1991, 1994, and 1978) and plot location, it was possible to produce five new sets of long-term *Trillium* densities and flowering percentages.

Table 2.4 – Summary of the primary researchers and research/field assistants involved in obtaining and entering data on *Trillium* densities and heights, from the commencement of data collection in 1991, to the most recent count completed in 2009.

	Primary Researcher(s)	***Research/Field Assistant(s); Data Input***
1991	Saewan Koh, Dawn Bazely	Harpreet Atwal
1992	Saewan Koh, Dawn Bazely	Harpreet Atwal, Tracy Tanentzap
1993	Saewan Koh, Dawn Bazely	Harpreet Atwal
1994	Saewan Koh, Dawn Bazely	
1995	Saewan Koh, Nancy Falkenberg, Dawn Bazely	Harpreet Atwal, Pratyusha
1996	Saewan Koh, Dawn Bazely	Tracy Tanentzap
1997	Saewan Koh, Kim Hynes , Dawn Bazely	Tracy Tanentzap
1998	Kim Hynes , Dawn Bazely	Tracy Tanentzap
1999	Kim Hynes , Dawn Bazely	Kim Hynes, Tracy Tanentzap
2000	Carrie Firanski, Dawn Bazely	Harpreet Atwal, Andrew Tanentzap, Pratyusha
2001	Carrie Firanski, Dawn Bazely	Andrew Tanentzap
2009	Dawn Bazely, Andrew Tanentzap, Saewan Koh	David Staples, Prasana S., Jelena Velic, Pratyusha, Tracy Tanentzap

2.1.3.1 Issues and conflicts involved in data synthesis and reconstruction

By involving a great number of researchers and field assistants in the collection of such measures, many discrepancies had arisen, as was found to be the case when comparing counts taken in identical plots, over a number of years. Each researcher had a unique way of responding to novel field problems that may be encountered in any given year. The re-establishment of certain plots in 1994, for example, required the movement of the control plot in either the north or south direction. Subsequent researchers, unaware of the intrinsic pattern involved, arbitrarily named the two controls which they encountered. This resulted in the label NC appearing inaccurately at many stations in which the north plot was moved, requiring that the new control be an SC. In this way, further investigative work was needed and attempts were made to reconstruct the thought process at both the time of establishment and at successive measures.

In addition, field notes of various years bore many researcher-specific abbreviations and notes, regarding the state of the forest at the time. The latter comments were essential in understanding the surveying methods. Many notes, however, as with the designation of the south and north controls, differed among individuals, making it difficult to decipher their true meaning. The lack of uniformity in both the field note sheets and computer spreadsheets clouded the process of integrating all of the data contained.

Another common issue was the mislabelling of the digital data, either by those who had initially entered the data or were allowed access to the files over the many years. Values filed away under the heading of 1994 were identical to those entered for 2000, which became even more confounding as the spreadsheet containing the former was labelled as "00 heights," yet bore the 1994 study dates. It was thus determined that the initial file containing 2000 densities was mislabelled, and the real sheet found later, mislabelled. A similar incident occurred with the 1995 data, for which several sets of data existed, each with a unique set of density values. In analyzing the height-to-leaf and height-to-flower measures in these spreadsheets and determining that flowering height was less than leaf height, it was determined that they were inaccurately attributed to *Trillium spp*., and actually referred to *Arisaema triphyllum* data. Both data sets, however, were formatted identically and had it not been for the presence of height values, it would have been impossible to distinguish one from the other. These issues stress the importance of quality control in data entry and analysis, particularly when working with values and files from a variety of different sources, over many years. For all data sets employed in this study, errors were present and the values verified and, in some cases, re-entered by Tracy Tanentzap.

2.1.4 Visualizing changes in *Trillium* densities, following deer culls and artificial exclusion

Following the cleaning and integration of these numbers, the selection of key data sets was necessary. Plots that never possessed any *Trillium*, over the entire time period, were removed and the remaining values assigned to a category of wet/mesic 1991 exclosures, wet/mesic 1994 exclosures, wet/mesic grazed plots, GEX/BEX, and Gardiner/Bennett grazed plots. It was possible to clump wet and mesic plots together, as the effects of specific soil type on *Trillium* growth were not being explored in this particular study, but rather the relative recoveries within the same set of plots, over years of varying deer densities.

An average *Trillium* density and standard error value were calculated for each of these five categories, in each year, and plotted on a column graph in Excel 2007. For W/M 1994 exclosures, average density values were only calculated from 1995 onwards, while the true Bennett and Gardiner sites, both exclosures and grazed plots, were not measured in 1994, 1995, and 2000. One-way ANOVAs were performed in SPSS version 19.0, on the density values of each category, over the entire range of years for which values were available. This determined among which years the differences in *Trillium* density were significant, at a level of $\alpha = 0.05$.

2.1.5 Fluctuations in percent flowering *Trillium*, following deer culls and artificial exclusion

In order to determine the percentage of flowering *Trillium* in each year, the number of *Trillium* individuals with an associated flower height was taken over the number of plants for which a leaf height was measured and available. This was then presented graphically using a column chart in Excel 2007.

2.2 Effects of deer exclusion on *Alliaria petiolata* and species richness

A second study on *Alliaria petiolata* presence and its effect on local species richness was also explored, simultaneous with the prior research on *Trillium* recovery, as a function of deer density control. This study was initiated in 1995, at both Rondeau Provincial Park and Point Pelee National Park, however only the former will be discussed, in the context of invasive spread following intense grazing by herbivores.

2.2.1 Historical plot setup for estimating spread of a non-indigenous plant species

In 1995, fifteen transects, 54 metres in length, were established at three sites within Rondeau Provincial Park (Figure 2.3; Figure 2.5): Ponybarn (6 transects), Lakeshore (6 transects), and Southpoint (3 transects). Along each transect, 19 quadrats were initially measured, each spanning 0.36 square metres. A twentieth quadrat was added to transects 1-3 at Southpoint, 1-4 and 6 at Lakeshore, and 2,4, and 6 at Ponybarn, in 2001.

A previous report looked at garlic mustard densities, in both their adult and rosette forms, as well as species richness, at all three sites (Bazely *et al.* 2009). The results of this study, however, excluded the results of a 2009 survey of species richness, and 2004 and 2009 garlic mustard densities, due to issues associated with quality control of data. In 2004, all 60 Southpoint quadrats were surveyed for garlic mustard densities, in both adult and rosette forms, as well as 113 Lakeshore quadrats and 52 Ponybarn quadrats. Species richness was not measured in this year. In 2009, 60 Southpoint quadrats were surveyed for overall species richness and garlic mustard counts, as well as 98 Lakeshore sites and 51 Ponybarn sites, with only transects 4 to 6 accounted for in the latter.

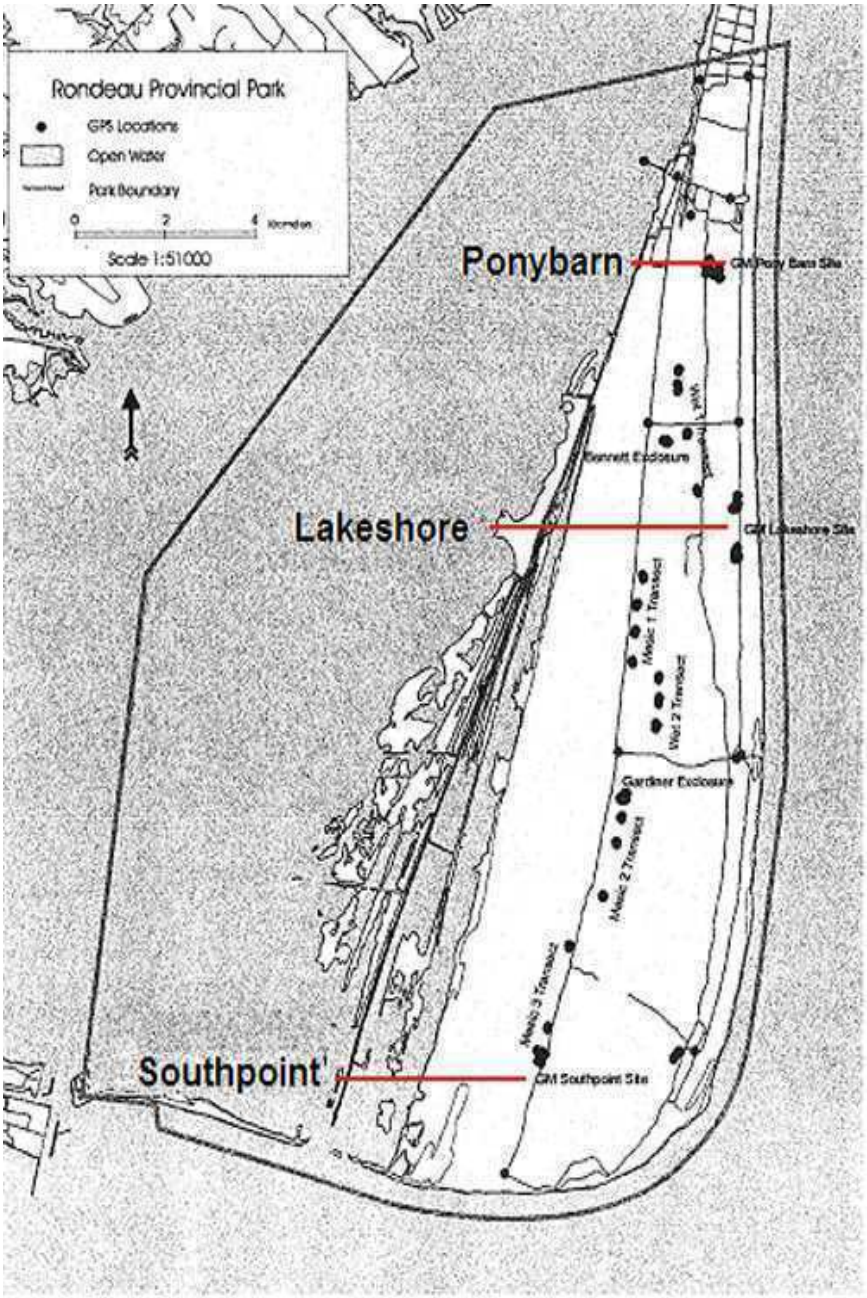

Figure 2.5 – Alternate map of permanent transects located at Ponybarn, Lakeshore, and Southpoint garlic mustard study sites at Rondeau Provincial Park (Firanski 2003).

2.2.2 Application of data synthesis and resulting analyses

In 2009, a report was presented to Rondeau Provincial Park officials, updating them on the research being conducted in the permanent plots throughout the park, with emphasis on analyzing the success of deer herd reductions on vegetation recovery (Bazely *et al.* 2009). In this way, the authors sought to characterize changes in plant species richness and *A. petiolata* spread at all three sites within the park, from 1995 to 2009. Garlic mustard densities, in both the rosette and adult forms, were included for 1995, 1996,

1997, and 2000. The most recent counts from 2004 and 2009, however, were not included. In addition, species richness values from 1995 and 2001 were featured, with the 2009 measures to be added. The report also featured a number of incorrect measures, due to erroneous entry of some 2009 data.

A survey of species richness was not conducted in 2004, and only the densities of *A. petiolata* rosettes and adults entered. Lastly, the problematic 2009 percentage species cover and density values, recorded incorrectly in the resultant spreadsheet, for all three sites, were eliminated and re-entered. Species richness values for 2009 were then extracted from these updated data summaries.

In fine-tuning Bazely et al. (2009), Table 1 of the paper had to be completed using the updated data. Due to the non-normal distribution of the data, and its resulting rightward skew, the geometric mean for each site, in each year, was calculated. This was done using only those plots that contained adult or rosette forms of *A. petiolata.* That is, in order to obtain the geometric mean rosette density, only plots in which rosettes were found, in that year, were counted and constituted the sample size, n. The geometric mean (Mn) was calculated as antilog of mean of (ln density). However, the geometric mean obtained for each plot corresponded to an area of 0.36 m^2 and, as such, had to be converted to a full 1.0 m^2. The number of plots containing rosettes and adults, respectively, at each site, for each year, was tallied. These values were then totalled across a year to provide the total number of plots containing each life form, for each year. In this way, the geometric means of rosette and adult densities in 2004 and 2009 were entered into the existing chart.

2.2.3 Assessing the temporal spread of *A. petiolata* and its consequent effects on species richness

A number of graphs present in the original report were also updated with the addition and revision of this data, most noteworthy a chart illustrating the percentage of surveyed plots containing garlic mustard, at each site, for the years 1995, 2001, 2004, and 2009. It was also desirable to determine whether garlic mustard presence affects species richness. Species richness values of all plots, in each year (1995, 2001, 2009), were divided on the basis of whether the respective plots contained garlic mustard in the original 1995 count. The species richness of these plots was then averaged over all sites, for both categories, and graphed.

In order to better elucidate the nature of the relationship among garlic mustard density and species richness, the latter was regressed on the former. By pooling 2009 data from all three sites, it was possible to gain a better understanding of the effect of garlic mustard density on community species diversity. Garlic mustard densities and measures of species richness, however, corresponded to a plot area of 0.36m^2. As a result, these values were converted to represent an area of 1.0m^2 before commencing a linear regression and bivariate correlation in SPSS version 19.0.

Chapter 3: Results

3.1 *Trillium* Recovery Following Deer Exclusion

In any one field season, *Trillium* counts were performed during the period between May and early July; in years in which several counts were performed, the maximum density was taken sities. As *Trillium* is a spring-flowering ephemeral, measures taken anytime from May to early July should be indicative of the entire emergent population for the given year, prior to early senescence.

3.1.1 Application of data synthesis principles and practices

3.1.1.1 Regressing percent *Trillium* cover on *Trillium* density values

Trillium density in each square metre plot was regressed onto *Trillium* percentage cover, per plot, in order to obtain density values for the 1999 year, in which only percentage cover measures were obtainable (Figure 3.1). Correlation analysis revealed a significant, positive relationship between the two variables (r=0.947, p=0.000; Table 3.1), while a regression analysis, fitted to a quadratic model, further revealed the corresponding graphical equation to be used in calculating 1999 densities ($y=-0.008+1.648x^2-0.138$; $R^2=0.902$; Table 3.2). This statistical output strongly supports the predictive value of percent *Trillium* cover in estimating *Trillium* density. Upon using the resultant equation to calculate 1999 *Trillium* densities, it was possible to construct a final *Trillium* density table for all surveyed years, using data synthesis approaches.

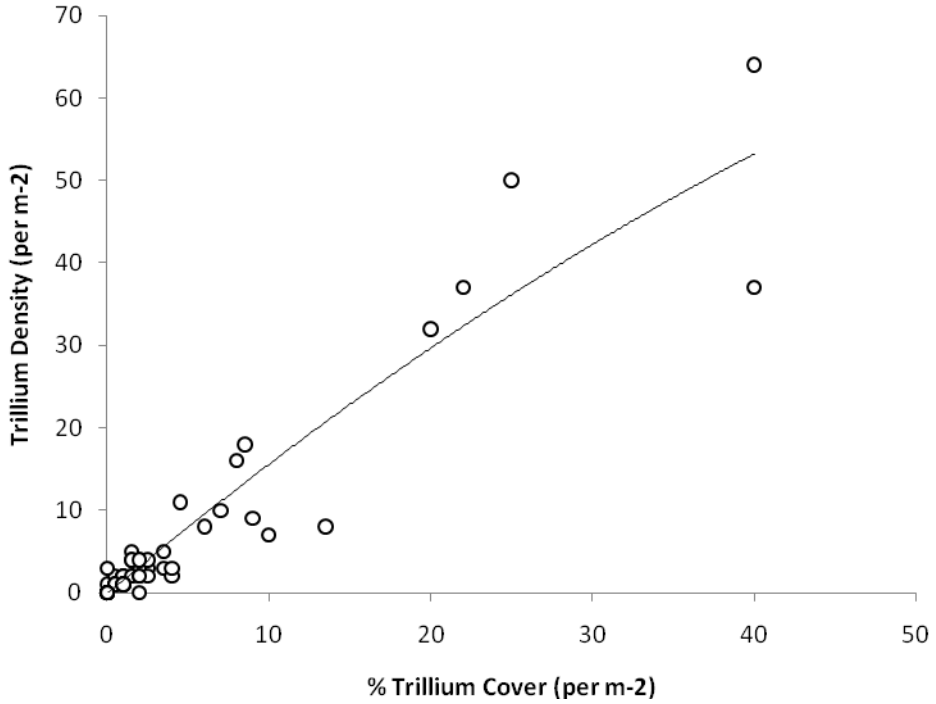

Figure 3.1 – Relationship between 2001 *Trillium* densities and percentage *Trillium* cover, per 1 m^2 plot ($R^2=0.902$).

Table 3.1 – Output table of a bivariate correlation performed on *Trillium* density versus percentage cover; a strong, positive relationship exists between the variables (r=0.947; p=0.000).

		PercentCover	Density
PercentCover	Pearson Correlation	1	.947**
	Sig. (2-tailed)		.000
	N	94	94
Density	Pearson Correlation	.947**	1
	Sig. (2-tailed)	.000	
	N	94	94

**. Correlation is significant at the 0.01 level (2-tailed).

Table 3.2 – Output table of a quadratic regression analysis performed on *Trillium* density versus percentage cover ($y = -0.008+1.648x^2-0.138$), for which a significant, positive relationship exists ($R^2=0.902$).

Dependent Variable:Density

	Model Summary					Parameter Estimates		
Equation	R Square	F	df1	df2	Sig.	Constant	b1	b2
Quadratic	.902	416.623	2	91	.000	-.138	1.648	-.008

The independent variable is PercentCover.

3.1.1.2 Results of data synthesis efforts

A final density table was produced for *T. grandiflorum*, at Bennett and Gardiner exlcosures and control sites, and wet and mesic exclosures and control sites in 1991, 1992, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, and 2009 (Appendix 1).

3.1.2 Visualizing changes in *Trillium* densities, following deer culls and artificial exclusion

Due in part to their long establishment and ongoing maintenance (since 1978), the Bennett and Gardiner sites serve as a backdrop and context for examining long-term trends in *Trillium* density (Figure 3.2). It was possible to detect the effects of deer culls in an area of the park that had experienced a longer period of forest recovery than the wet and mesic exclosures. That is, the exclosures at these sites have been subject to decreased deer densities since their establishment in 1978, making it possible to observe changes in *Trillium* density, and subsequently forest restoration, over a longer period, one which may be more reflective of recovery times required in a forest that has experienced chronic high deer herbivory. The results of a one-way ANOVA revealed that there were no significant differences among the different years, for each station type (grazed,

exclosure). *Trillium* density fluctuated from 2 to 8 plants m^{-2} inside the exclosures and around 1 plant m^{-2} outside. This suggests that external *Trillium* populations continue to be affected by deer herbivory.

Trillium densities were categorized based on the soil and exclosure type in which they were measured, and averaged among all sites that met this criteria, for each survey year. These divisions included wet/mesic grazed plots, the initial 1991 wet/mesic exclosures , and the re-established 1994 wet/mesic exclosures. The average *Trillium* density in each category was then plotted, for each year, with standard error displayed (Figure 3.3). Additionally, a one-way ANOVA was run on the densities within each category, with the surveying years acting as different treatment types. Within the 1991 exclosures, no significant differences in *Trillium* density were evident, over all of the years of data, while the grazed exclosures expressed a significant decrease in density from 1995 (n=14) to 1996 (n=20; p=0.009; Appendix 2) and 1995 to 1997 (n=20; p=0.013; Appendix 2). A similar pattern was evident in the 1994 exclosures, whereby the 1995 densities (n=11) varied significantly from those lower values measured in 1998 (n=9; p=0.047; Appendix 3) and 2001 (n=14; p=0.032; Appendix 3). However, overall, there were increasing trends in *Trillium* densities in the smaller deer exclosures compared with control, grazed areas, but also highly fluctuating densities. In the exclosures, the densities were in the same range as in Bennett and Gardiner exclosures. The increased densities in grazed areas during the mid to late 1990s, appeared to have declined during the last decade.

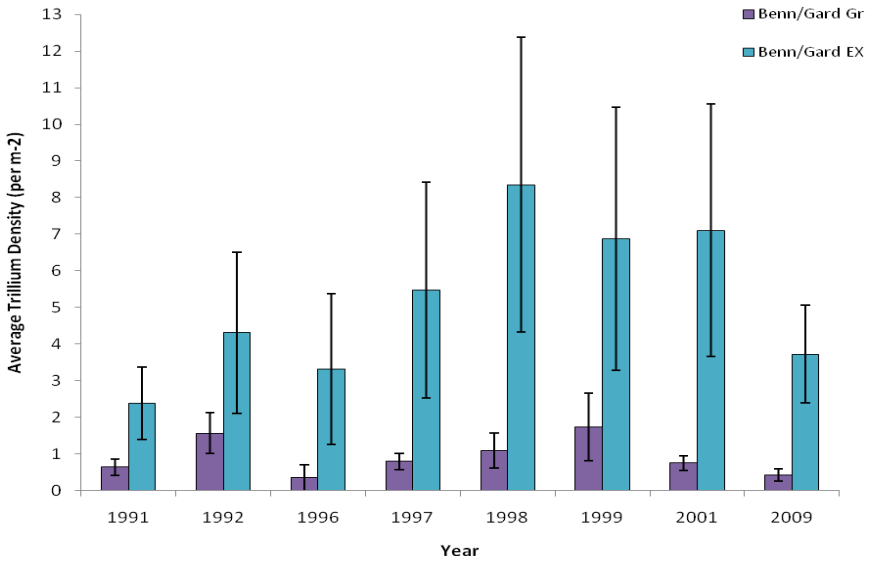

Figure 3.2 – *T. grandiflorum* densities, plants per 1 m^2, at Bennett and Gardiner exclosures and adjacent grazed plots, from 1991 to 2009, with standard error of the mean presented.

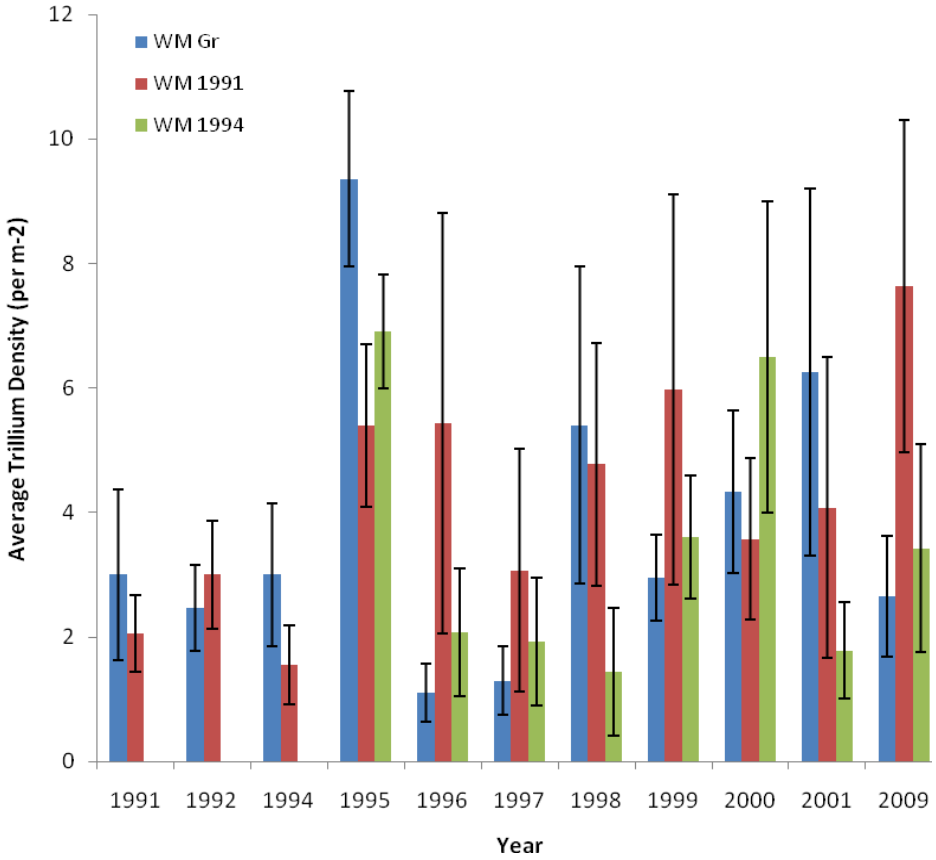

Figure 3.3 - *T. grandiflorum* densities, plants per 1 m^2, at wet and mesic exclosures (1991 and 1994) and grazed plots, from 1991 to 2009, with standard error of the mean presented.

3.1.3 Fluctuations in percentages of flowering *Trillium* plants, following deer culls and artificial deer exclusion

In order to track the recovery of *T. grandiflorum*, a measure of the population's reproductive ability is needed. By tracking the percentage of flowering *Trillium* in each plot, it is possible to determine whether individual *Trillium* are recovering to the point of reaching the minimum reproductive height, and whether densities can be expected to increase as a result of more flowering individuals (Figure 3.4). Two years following the establishment of the original 1991 exclosures, the percentage of flowering *Trillium* began to increase at these sites, reaching an all-time high of 50% in the most recent study year, 2009. A similar pattern is evident in the grazed sites, whereby the percentage of flowering *Trillium* has increased from 0, in three consecutive years, to a 2001 value comparable to that of the1991 exclosures. The Bennett and Gardiner exclosures display a much greater percentage of flowering *Trillium*, as do their grazed plots, which have seen a continual increase over the years.

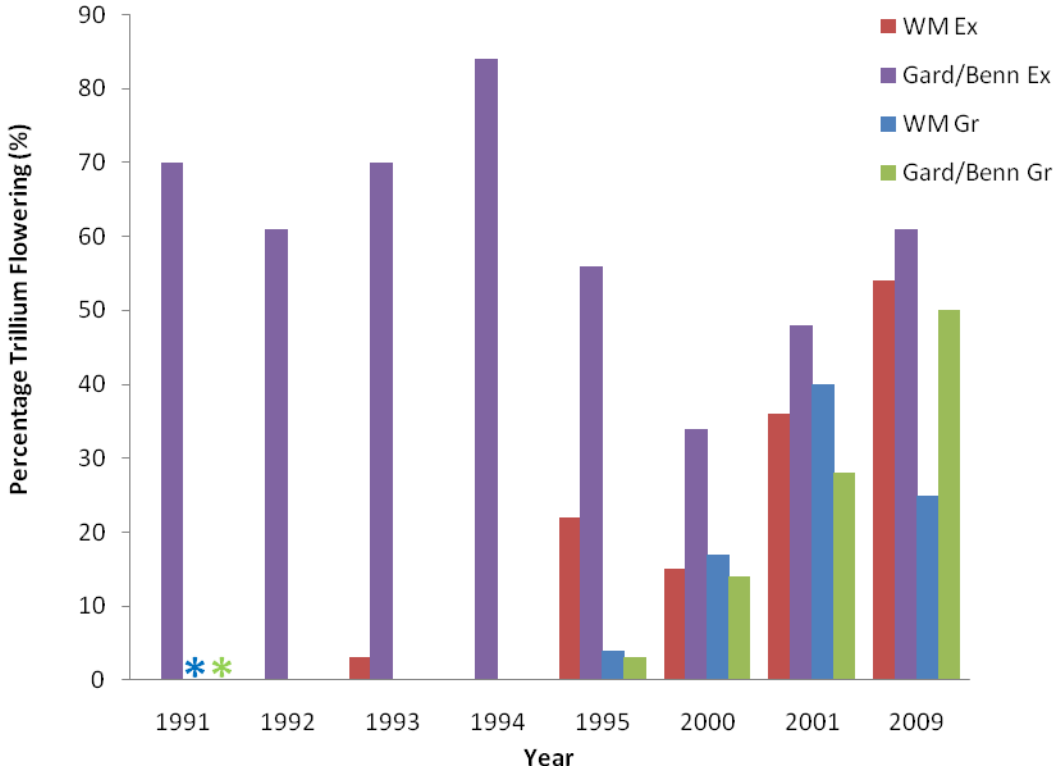

Figure 3.4 - Relative percentage of flowering *T. grandiflorum* individuals, per 1 m^2 plots at Bennett and Gardiner exclosures and grazed plots, and wet and mesic exclosures and grazed plots, from 1991 to 2009, with asteriks representing categories that were not sampled in that year.

3.2 Effects of deer exclusion on *Alliaria petiolata* and species richness

3.2.1 Application of data synthesis and resulting analyses

A final table was produced for *A. petiolata* densities and overall species richness, at Southpoint, Lakeshore, and Ponybarn for 1995, 2001, 2004, and 2009 (Appendix 4). These values were further manipulated to complete a summary chart of garlic mustard adult and rosette densities (Appendix 5).

3.2.2 Assessing the temporal spread of *A. petiolata* and its consequent effects on species richness

A graph featured in Bazely et al. (2009), expressing the proportion of plots containing garlic mustard, in each year, updated to include accurate data (complying with quality control criteria) compiled from the 2009 count, is presented here (Figure 3.5). Although there was an increase in garlic mustard presence between 1995 and 2004, for both Southpoint and Ponybarn, this measure declined leading into 2009. Lakeshore, however, seemed to fluctuate among years, with no discernable pattern evident. Despite a slight increase in garlic mustard presence at both Southpoint and Lakeshore, from 1995 to 2009, Ponybarn appears to have decreased in the proportion of garlic mustard-containing plots when compared to its 1995 levels. It is evident from these graphs that *A. petiolata*

is not spreading, as was anticipated; furthermore, it is not the dispersal of garlic mustard itself that is of primary interest, but rather, its potential interaction with and effect on community species composition.

Species richness counts from all three sites were pooled, to produce an average value for each year, corresponding to whether or not the plots in question contained garlic mustard in 1995, the year of establishment (Figure 3.6). From 1995 to 2001, species richness has increased at both sites: those that did and did not contain garlic mustard. In fact, by this point, garlic mustard-containing plots expressed one more species, on average, than their non-invaded counterparts. By 2009, however, species richness counts had declined, but remained above 1995 levels. This decline in species richness corresponds to a period in which the proportion of plots containing garlic mustard had also declined, a uniform pattern seen at all three study sites (Figure 3.5).

In order to better understand the effect that garlic mustard presence and corresponding density is having on species richness, a regression was performed. Both species richness and garlic mustard densities were standardized to correspond to an area of one square metre, and data from all sites were pooled together for the 2009 year. A regression was then plotted (Figure 3.7) and a bivariate correlation performed (Table 3.3). It was determined that there is a weak, positive correlation between garlic mustard density and species richness ($r=0.165$). The results of the corresponding regression revealed that little variability in species richness may be attributed to garlic mustard densities ($R^2=0.027$; Table 3.4). This slight, positive influence of garlic mustard on species richness is also not significant ($p=0.100$; Table 3.5). Thus, garlic mustard densities cannot reliably be used to predict species richness.

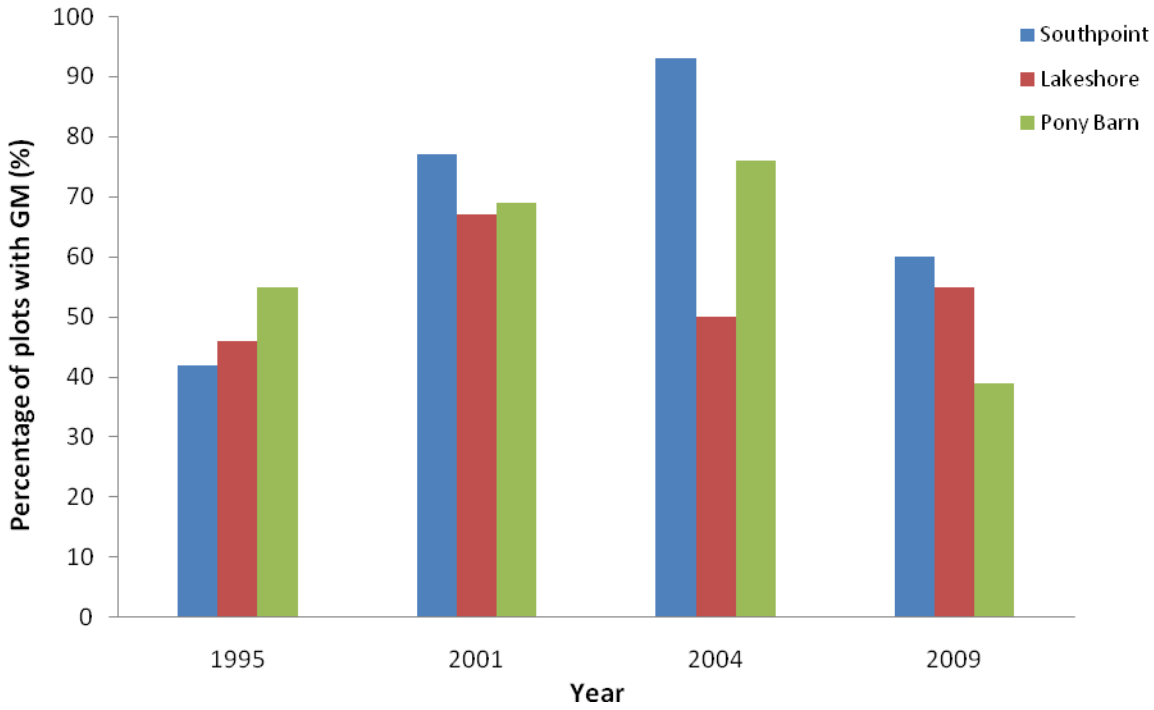

Figure 3.5 - Percentage of plots containing *A. petiolata* individuals, at each site, from 1995 to 2009.

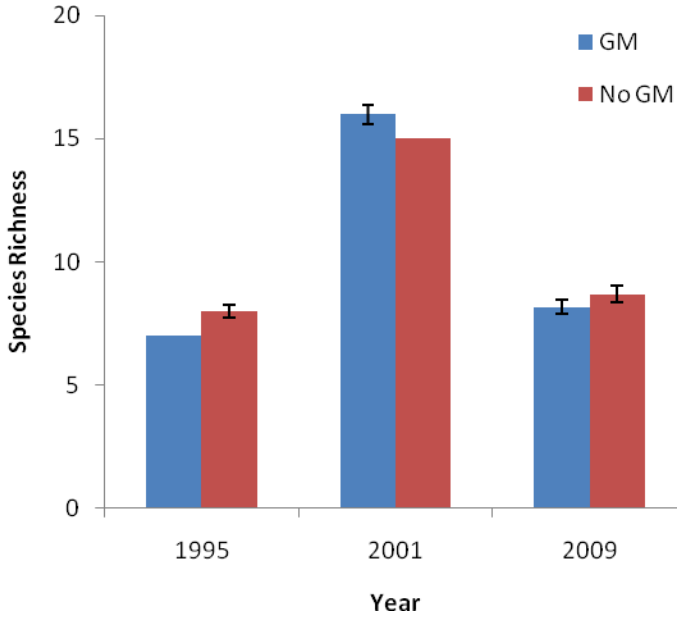

Figure 3.6 – Relative species richness for all sites, distinguished by garlic mustard presence at the onset of the study in 1995.

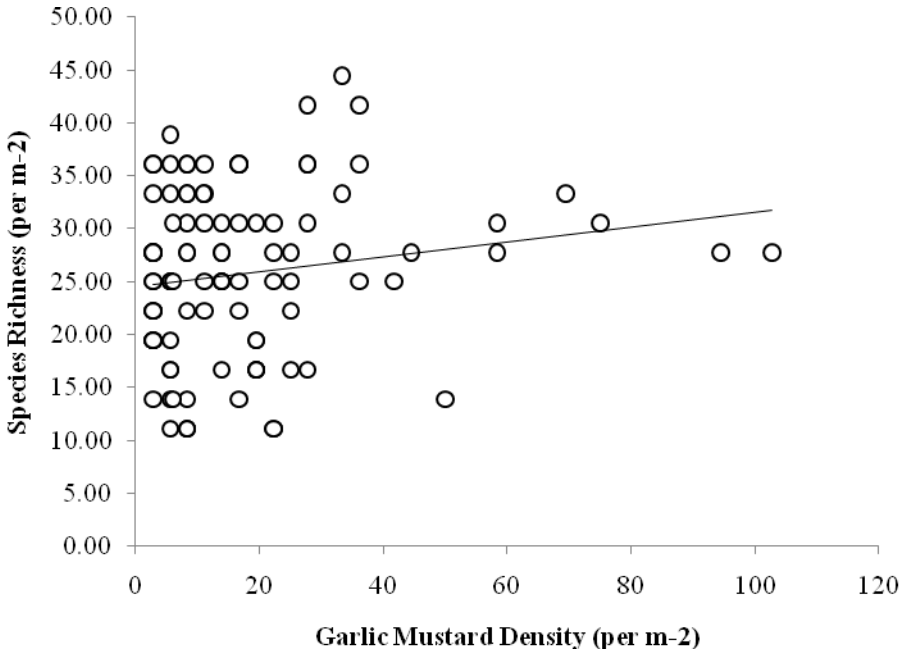

Figure 3.7 – Weak, positive relationship between 2009 species richness and *A. petiolata* density, per 1 m^2 plot (R^2=0.027).

Table 3.3 - Output table of a bivariate correlation performed on species richness versus garlic mustard density; a weak, positive relationship exists between the variables (r=0.165; p=0.100).

		GMDensity	SpeciesRichness
GMDensity	Pearson Correlation	1	.165
	Sig. (2-tailed)		.100
	N	101	101
SpeciesRichness	Pearson Correlation	.165	1
	Sig. (2-tailed)	.100	
	N	101	101

Table 3.4 – Output table of a linear regression analysis performed on species richness versus garlic mustard density, for which a weak, positive relationship exists (R^2=0.027).

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.165[a]	.027	.017	7.87907

a. Predictors: (Constant), GMDensity

Table 3.5 - Coefficient output of linear regression for 2009 species richness versus garlic mustard densities, illustrating a non-significant relationship between the variables (p=0.100).

Coefficients[a]

Model	Unstandardized Coefficients		Standardized Coefficients	t	Sig.
	B	Std. Error	Beta		
(Constant)	24.529	1.095		22.399	.000
GMDensity	.071	.042	.165	1.660	.100

a. Dependent Variable: SpeciesRichness

Chapter 4: Discussion

4.1 Evaluating the use of data synthesis principles and practices

The study sought to address specific research objectives, while simultaneously evaluating the usefulness of data synthesis methods in long-term ecological research. The application of historical data for purposes that may not have been anticipated by the original field researchers helps to resolve the "bottom drawer" issue of scientific research: what is to be done with the plethora of data lurking within the depths of academics' closets? Only a small percentage of collected data are actually published, and many academics and graduate students leave behind yellowing field notes and decaying binders, never to be looked at again, but which may potentially contain a goldmine of information. The establishment of the NCEAS helped to resolve this issue for many ecological researchers. By creating a digital repository of excess data, to be used by fellow academics, the field now became one of *"no datum left behind"*.

The value of synthesizing old data sets has been discussed in several papers (Ellison et al. 2006; McIntosh et al. 2007; Ellison 2010). Obtaining such data, however, may prove to be the most difficult feat of all. Researchers can spend countless weeks tracking down old records, only to find statistical summaries or graphs in their place (Ellison 2010). Raw data in the form of original field notes or starting digital files thus prove invaluable in synthesis efforts. Judgment calls are involved at nearly every step of the data synthesis process, from the formation of a study question to determining which data should be included and excluded. In instances where proper records are kept of the data selection process and consequent analyses (i.e. process metadata), ecological synthesis has proven successful (Ellison et al. 2006; Ellison 2010). There are, however, some studies in which results appear irreproducible, even when accompanied by such detailed methodology. This reflects differences in individual interpretation and manipulation of the synthesized data, rather than issues inherent in the novel data (Ellison 2010). Ellison (2010) stresses that a lack of reproducibility as a result of differing thought processes, and consequent analysis, does not delegitimize the data synthesis process or reduce its value in ecological studies. The primary problem with data synthesis studies, however, is the lack of accompanying, detailed process metadata, an issue that may be resolved by the introduction of analytical webs which would serve to illustrate the thought process involved in such syntheses (Ellison et al. 2006). Clearly, these are next steps that should be considered for the data in this study, that build upon the work I have done, along with proper archiving and organization of field notes. For example, Dr. Bazely is convinced that the missing 1999 field data are somewhere in a box!

The similarity of my experience in this research and in the study's findings, to those of comparable literature, further reinforces the validity of data synthesis in ecological reconstruction. By replacing current observations with a long-running set of historical data, similar conclusions may be reached without so much as leaving the privacy of one's virtual world.

4.2 *Trillium* Recovery Following Deer Exclusion

4.2.1. Visualizing changes in *Trillium* densities, following deer culls and artificial exclusion

Despite all of the research performed on *T. grandiflorum* in Rondeau Provincial Park, none has focused on changes in density as a result of deer culls and artificial exclusion. By creating a novel data set from unused observations of the past eighteen years, it was not only possible to address a new study question, but to also complement the results of a manuscript by Koh et al. examining *Trillium* height recovery, that has been submitted and rejected. A criticism of this study was that it failed to demonstrate the recovery of entire *Trillium* populations, rather than individuals (D. Bazely, personal communication). These results help to resolve this issue, while also introducing a key, missing piece of the *Trillium*-deer puzzle that has been the subject of much research by D. Bazely and her students over the past twenty years.

In the long-running Bennett and Gardiner exclosures, there is a consistently greater *Trillium* density than in the corresponding control plots in which deer grazing was allowed to continue, suggesting that artificial exclusion methods are effective in aiding *Trillium* recovery (Figure 3.2). A 2001 park management report similarly noted that annual deer culls had failed to make the vegetation of grazed plots more similar to these long-term exclosures (Ontario Parks 2001).

In the 1991-established exclosures, however, there is a great deal of observable fluctuation in density levels (Figure 3.3). Although increases are evident, none are significant, even when comparing 1991 values to those of 2009, by which point densities have increased, on average, almost four-fold. The 1994 exclosures display a slightly different pattern, whereby only a year after their establishment, relatively high *Trillium* densities were noted. This value quickly drops in the following year, from which point on there are annual fluctuations in deer density. These results appear to suggest a lack of a stable state for *Trillium* densities within this period.

The control plots tell a different tale, one of the efficacy of non-artifical exclusion methods: deer culls. The greatest *Trillium* density was observed in 1995, a spike that is best attributed to the 1993 cull. However, of the *Trillium* sampled in 1994, none were found to be flowering (Figure 3.4). How, then, is this increase in density, noted in 1995, possible? It is likely that some *Trillium* individuals were flowering elsewhere, and their seeds carried within the confines of these grazed plots, helping to increase densities. However, 1995 also proved to be a troublesome year in the construction of the newly synthesized data sets. The significant drop in density, from 1995 to 1996 and 1997, is most likely due to the cessation of these culls, as an increase is once again evident in 1998 onwards, following the re-commencement of annual culls the previous winter.

Recovering populations are also encumbered by the feeding preferences *of O. virginianus*; namely, their selectivity for larger *T. grandiflorum* individuals (Knight 2003, Koh et al. 2010). Given that reproductive status in *Trillium* is determined by height, grazed populations will not only be shorter, but nonreproducing and will thus require several years in order to reach the flowering stage, at which point individuals may contribute to proliferation (Anderson 1994, Kirschbaum and Anacker 2005, Koh et al. 2010).

Although culls helped to increase densities in grazed plots, naturally-occuring fluctuations were still evident. These results suggest that recovery in such disturbed *T. grandiflorum* requires a substantial period –in this case, more than eighteen years –in order to demonstrate significant recovery, a notion echoed by similar studies of forest restoration at Point Pelée National Park (McLachlan and Bazely 2001; McLachlan and Bazely 2003).

4.2.2. Fluctuations in percent flowering *Trillium*, following deer culls and artificial exclusion

In similar studies limiting overgrazing, increased flowering rates and leaf area were evident after only two years (Augustine and Frelich 1998). Such a pattern is evident within the combined 1991 and 1994 wet and mesic exclosures (Figure 3.4). There is a dramatic increase in percentage flowering *Trillium* from 0 in 1991, 1992, and 1994, to more than 50 in 2009, the most recent year of study, with intermittent recovery evident in 1993 and 1995. The Bennett and Gardiner exclosures have expressed a greater percentage of flowering *Trillium* than their grazed plots over the years, although the latter has seen a continual increase. These exclosures indicate that recovery of *Trillium* populations *is* possible, given a sufficient period of deer exclusion. In 1991, thirteen years after their establishment, almost 70% of *Trillium* within these exclosures are flowering, compared to 0 in the newly established wet and mesic exclosures, over the first two years (1991-1992).

In the wet and mesic grazed plots, for which deer culls are the primary factor controlling *Trillium* recovery, no *Trillium* is found to be flowering between 1992 and 1994, but by 1995, an increase is evident. The Gardiner and Bennett control plots, similarly affected by deer culls, also begin to see an increase in flowering *Trillium* in 1995, a mere two years following resumption of such practices. The lack of any flowering *Trillium* for both sets of grazed plots, from 1992 to 1994, may be attributable to the lack of culls a few years prior to this period. The cull of November, 1993 may then be responsible for the commencement of an increase in flowering *Trillium* at these grazed sites, two years later. Previous research found less *T. grandiflorum* than expected, at restored sites, due to the species' reliance on ant dispersal and the poor competitive ability of spring ephemerals, compared with ruderal species (McLachlan and Bazely 2001). However, the lag between deer culls and flowering of *Trillium* individuals may be responsible for the slow, inconsistent increase in densities. With the resumption of flowering individuals within the park, a seed bank for *T. grandiflorum* and other

vulnerable spring ephemerals, will deposit and allow for colonization of newly-recovering areas (Ontario Parks 2001).

4.3 Effects of deer exclusion on *Alliaria petiolata* and species richness: assessing the temporal spread of garlic mustard and its consequent effects on species richness

The addition of accurate 2009 data helps to strengthen the case made by Bazely et al. (2009), in that garlic mustard populations are not spreading across Rondeau as rapidly as previously believed. By illustrating that the proportion of garlic mustard within the Southpoint and Ponybarn experimental sites did indeed decrease from 2004 to 2009, with Ponybarn levels even falling below those of 1995, concern over the spread of this invasive species may be alleviated (Figure 3.5). Despite the fact that in 2004, nearly 100% of Southpoint plots contained garlic mustard, this value was actually able to drop dramatically by 2009, suggesting that the spread of this species is not as intensive as it has been made out to be. Furthermore, the minimal change in garlic mustard presence from 1995, when no cull occurred the previous winter, and 2009, following almost 12 years of annual culls, suggests that its spread may not be reliant on deer herbivory. The majority of research on *A. petiolata* has focused on its allelopathic effects on neighbouring plants, research which was rarely conducted in a naturally occurring field environment. As a result, much of the fear of garlic mustard spread relates to its perceived potential to lower species diversity of an invaded area (REF). Thus, its spread is of minimal concern if it is not proven to inhibit the productivity of the overall plant community.

In examining species richness at the same three sites used for *A. petiolata* monitoring, it was determined that species richness expressed little variation since the commencement of research in 1995 and that these sites now possess, on average, one additional species than at study onset(Figure 3.6). In comparing the effects of garlic mustard presence on species richness, it was found that presence of this invader had little to no effect on diversity, insofar as to even have *A. petiolata*-afflicted plots express one more species than their non-invaded counterparts, in 2004. The drop in species richness, between 2004 and 2009, corresponds to a period in which garlic mustard densities declined. The pattern of overall species richness and garlic mustard presence are very similar, suggesting that something else is controlling the growth and diversity of plants in the park.

In order to further elucidate the relationship between garlic mustard presence and species richness, the latter was regressed onto *A. petiolata* density, for all plots in 2009 (Figure 3.7). The slight, positive relationship between the two variables indicates that increasing garlic mustard density is not inhibiting productivity of neighbouring plants. Although it can be confidently stated that garlic mustard is not the *cause* of greater species richness, as may be extrapolated from the positive, linear relationship, it can be said that garlic mustard is not negatively affecting species richness.

Chapter 5: Conclusions

Côté et al. (2004) have proposed the notion of a point of no return, beyond which simply lowering deer densities is ineffective. In order for disturbed system to recover, human intervention and active reintroduction of imperilled species must be coupled with sustained deer densities (McLachlan and Bazely 2001; Côté et al. 2004). Such a management approach is advisable for Rondeau Provincial Park, where *T. grandiflorum* recovery is still underway, at a predictably slow rate. After thirteen years of annual deer culls, the park is still in the process of recovery and will continue to suffer from low trillium densities for years to come. Fears about *A. petiolata*, however, may be alleviated with evidence of containment of their spread in recent years and no overall decline in species richness due to its allelopathic abilities. The results of such a long-term study, using data synthesized from various research projects conducted over the past eighteen years, confirm the risk involved in forming management policies based on brief studies of intermittent periods.

Additionally, these results have demonstrated that a data synthesis was useful. The efforts undertaken in this thesis should be continued to make the data available in a digital repository.

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